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RESEARCH ARTICLE

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Fire response of the endangered *Pinus culminicola* stands after 18 years in Cerro El Potosí, northeast Mexico

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Abstract

Aim of the study: To analyze the impact of the 1998 wildfire on dwarf pine (*Pinus culminicola*) population stands 18 years after the fire.

Area of study: Cerro el Potosí, Nuevo León (México).

Material and methods: We measured regeneration, biovolume of individuals, diversity and species composition in unburned and burned plots randomly selected in the area.

Main results: Recovery was very slow, with limited dense natural regeneration and very slow growth. Significant differences between control and burned plots were found with respect to regeneration, species composition, evenness and soil nutrients, suggesting that fire could promote natural regeneration and biodiversity in these communities.

Research highlights: Management efforts are necessary to maintain a natural fire regime and ensure grazing exclusion in order to favor the conservation of the community dominated by this rare species *P. culminicola*.

Additional keywords: fire ecology; DCA; Permanova; richness.

Abbreviations used: CIC (Cationic Interchange Complex); DCA (Detrended Correspondence Analysis); EC (Exchangeable Cation); OM (Organic Matter).

Authors´ contributions: Conceived, designed and performed the experiments: JRA, EE and HGR. Analyzed the data: JRA, JAV, JRE and YM. Contributed reagents/materials/analysis tools: JAV, JRE, YM, IC and JU. Wrote the paper: JRA, JAE and EE. All authors read and approved the final manuscript.

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Introduction

Habitat loss and invasive species are the main threats to the survival of most endangered species (D'Antonio & Kark, 2002), especially for species with small populations and slow regeneration. This is common at high altitudes due to the limited productivity of these sites and the short growing season (Mark & Baylis, 1963; Whittaker & Fernández-Palacios, 2007). If these species have also been reduced to just small remnant populations, natural processes such as fire can mean a dangerous threat of extinction. In addition, there are many other elements of change that can endanger these populations (Walker & Steffen, 1997) in the long term, such as global warming, which can be intensified in these small and environmentally limited populations (Farjon, 2013).

Many high elevation forest ecosystems are adapted to stand-replacing fires (Baker, 2006), and fire can be a key disturbance process influencing tree survival and regeneration (DeBano *et al*., 1998). The recovery of a stand will depend on seed production and dispersion (Donato *et al.,* 2009), and climate conditions will govern the success of seedling and sapling establishment (Arévalo & Fernández-Palacios, 2008; Allen *et al.,* 2012), with post-fire conditions playing an important role.

Dwarf pine (*Pinus culminicola*) was described by Andresen & Beaman (1961) as a new species from the highest ridges of Cerro El Potosí, Mexico. In 1966, the species occupied a total area of 106 ha (Beaman & Andresen, 1966), and a few years later, it was found in Sierra La Marta and Sierra La Viga in the state of Coahuila, and Cerro Peña Nevada in Nuevo León, with spatial patterns and altitudinal distribution ranges similar to those determined for Cerro El Potosí (Riskind & Patterson, 1975). Fire is a key component of the dynamics of these communities, and in 1978, wildfires on Cerro El Potosí burned 34% of the area occupied by dwarf pine (García, 1989). Other anthropogenic impacts have also reduced the area formerly covered by dwarf pine, including road building, cattle grazing, mining and land use changes for the installation of telecommunication and aerial navigation infrastructure (Jiménez *et al.,* 2005).

In 1998, several wildfires drastically reduced the area covered by dwarf pine on Cerro El Potosí. Currently, only 30 ha of severely fragmented dwarf pine area exist in Cerro El Potosí, including adults with low seed production that are subject to the impact of grazing and trampling by cattle and horses (Jiménez *et al.,* 1999). There are no studies of *P. culminicola* populations on other mountains where it occurs, and unfortunately most of these areas are not legally protected and are also subjected to grazing and human-induced fires.

This is a species with very small distribution range, highly sensitive to fire, whose population has been reduced and is endangered. Furthermore, it grows in an area where environmental conditions, including drought and very low temperatures, lead to a short growing season. Therefore, changes in the natural disturbances patterns could result in the extinction of the species.

The main objective of this study was to study the regeneration, almost 20 years after the last fire affected this small stand. We tested the hypothesis that *P. culminicola* follows a pattern of stand replacement after fire and that its regeneration requires fire in order to renew individuals in the area.

Material and methods

Study area

The study area is located at the top of the Cerro El Potosí. This is a legally protected natural area, established by the government of the state of Nuevo León under the category of ecological conservation (PO, 2000). This mountain is located on *ejidal* and private lands in the south of the State of Nuevo León, Mexico (lat. 24°52´N, long. 100°13´W) within the Sierra Madre Oriental range, between the Gulf Coastal Plain and the Mexican High Plateau (Fig. 1) and dates in some parts from the inferior cretaceous with a marine sedimentary origin (García & González, 1991). Soils are classified as Orthens and also some Rendolls with poor discrimination of horizons, as well as Litosoils (González, 1999). The maximum elevation at Cerro El Potosí reaches 3,721 m.

The climatic data indicate that it is an E(T) H(e) type climate following the Köppen classification modified by García (1988). Mean annual precipitation is low, less than 600 mm, falling mainly in the month of September and supplemented by fog condensation. Average annual temperature is 12 ^{o}C, with an average minimum in January of -1.5 ^{o}C. The mountain top is exposed to high winds for most of the year. Dense fogs often occur throughout the year. Snow may fall several times during winter, persisting for a few days on northern and northeastern slopes (García, 1996).

In this area, the population of *P. culminicola* occurs mainly over 3,600 m a.s.l. with isolated individuals below that altitude. Density is highly variable ranging from 200 to 25 ind/ha. Above 3,600 m, it is the dominant shrub, forming shrublands associated with *Juniperus zanonii* interspersed with occasional gnarled and stunted *Pinus hartwegii* (Estrada *et al.,* 2014). This community was described by Rzedowski (2006) as *Pinus* shrubland and occupies a narrow altitudinal belt, growing between the *Pinus hartwegii* forest and the alpine meadow.

The species

Pinus culminicola Andresen & Beaman is a dwarf shrub 0.5-2.5 m tall and with more than 1.0 m of canopy cover. It is multi-stemmed with low, dense and rounded crowns. The bark is grey-brown, thin and scaly at the base of the trunk. The leaves (‘needles’) are in fascicles of five, slender, 3-5.5 cm long, and deep green to blue-green, with stomata confined to a bright white band on the inner surfaces. The cones are globose, 3-4 cm long and broad when closed, green at first, ripening yellow-brown when 16-18 months old, with only a small number of thin, fragile scales, typically 6-14 fertile scales. The cones open to a breadth of 4-6 cm when mature (non-serotinous), approximately in October, holding the seeds on the scales after opening. The seeds are 9-12 mm long, with a thick shell, a white endosperm, and a vestigial 1-2 mm wing (Estrada *et al.,* 2014). They are dispersed by the Clark’s nutcracker, which pluck the seeds out of the open cones. These jays, which use the seeds as a major food resource, store many of the seeds for later use; however, some of these stored

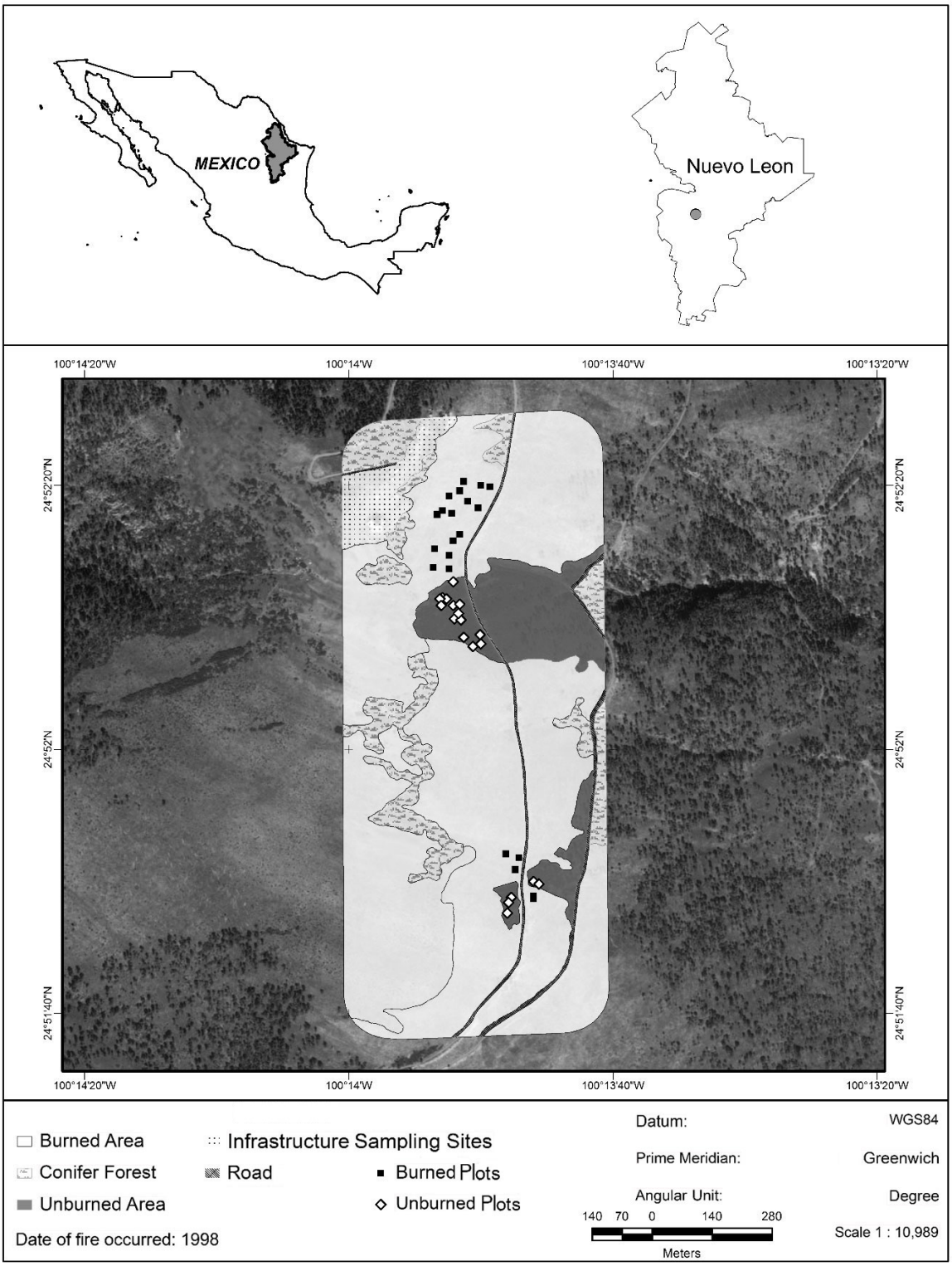

Figure 1. Location and map of the studied area in Cerro El Potosí, Mexico. Burned and unburned plots are indicated.

seeds are not used and are able to grow into new plants. The species is endemic to a few mountain tops in the states of Coahuila and Nuevo León in northeast Mexico (Villaseñor, 2016). The conservation status of the species according to SEMARNAT (2010) is in danger of extinction. Following the IUCN Red List of Threatened Species, this species is assessed as endangered under criterion B1+2 (restricted distribution to few populations), and grazing and fire are threatening these populations (Farjon, 2013). *P.*

culminicola has a very restricted range, and while not subject to intensive cutting for fuel wood, it is, during long dry periods, very susceptible to fire (Perry, 1991).

Sampling design

During October-November 2015, we sampled 40 plots in the Cerro El Potosí, in the area with the densest existing population, affected by fire in 1998 and 1978, both of which were considered intense and severe fires during summer periods (García & Gonzalez, 1991). In the burned area (hereafter burned), we randomly located 21 plots, while in the unburned area (hereafter unburned) we located 19 plots (Fig. 1). Each plot was 15 × 15 m (225 m^2).

In each plot, we collected one composite soil sample of about 1 kg from 0-30 cm depth. Each sample was mixed, dried, and sieved through a 2 mm sieve, and debris and stones were removed from the soil. We analyzed pH, EC (electrical conductivity of extract, dS/m), Olsen P (ppm), N (ppm), % organic matter (OM), available cations in ppm (Na, K, Ca, Mg) and CIC (Cationic Interchange Complex, meq/100 g). Percentages of clay, sand and silt were also analyzed. We followed standard methods of analyses (AOAC, 1990; MAPA, 1986). Altitude, aspect and slope were measured in each plot.

The available information on fires indicated that in 1978, 30% of the pine population was burned, while in 1998, after several fire events, the population was reduced to 30 ha of fragmented patches. The fire intensity was high enough to eliminate all the individuals' affected (Jiménez *et al*., 2005). In the burned plots, all regeneration woody plants (trees and shrubs) after 1998 were recorded together with the height and two crown diameters (the larger one and its perpendicular), which allowed for the calculation of the total biovolume of the species (Arévalo *et al*., 2015). Dead trees were also counted and measured. In the unburned plots, due to the dense cover of the pines, only the total cover of pines and other woody species on each plot were recorded, and regeneration was noted (individuals under 130 cm tall).

All species cover on plot surfaces was estimated and noted on a scale of 1 to 9 (cover classes: 1, traces; 2, >1% of cover in the plot; 3, 1-2%; 4, 2-5%; 5, 5-10%; 6, 10-25%; 7, 25-50%; 8, 50-75%; 9, >75%). A basic feature of biological plant communities is the distribution of abundance among species, so we calculated the Smith & Wilson (1996) evenness index for each plot. An estimation of rock, soil and litter cover was made in each plot using a scale of 1 to 9 (the same scale as used for species cover).

For species names, we followed the checklist of flora for the summit of Cerro El Potosí (García & González, 1991).

Statistical analyses

We performed a one-way distance-based permutational t-test (Anderson *et al.,* 2008) for comparison of species richness and Smith & Wilson evenness values of unburned plots *vs*. burned plots. The analyses were based on Brian-Curtis similarity of the raw data, with *p*-values ($p < 0.05$) obtained with 9999 permutations and a Monte Carlo correction when necessary. Primer 6 and Permanova+ (PRIMER-E Ltd, Plymouth, UK) were used to perform all PERMANOVA statistical procedures.

We used DCA (Detrended Correspondence Analysis; Hill & Gauch, 1980) to analyse the species composition (based on species cover) of burned *vs*. unburned plots. The coordinates on the first DCA axis for each plot were correlated with the environmental and soil nutrient variables using the Pearson correlation coefficient and tested for significance ($p< 0.05$) and the multiple test procedure of Holm for independent tests (Legendre & Legendre, 1998) was also applied.

For the significantly correlated variables with the coordinates of the plots, we performed a one-way distance-based permutational t-test as before. To determine if DCA axes discriminated the species composition based on burned *vs*. unburned, the coordinates of transects from the DCA for axes I and II were analyzed using logistic regression and the Wald statistic to determine its significance (for $p<0.05$). We performed all multivariate analysis with the CANOCO package (ter Braak & Ŝmilauer, 1998). The statistical methods used followed Zar (1984) and were implemented using Primer 6 and Permanova+ (PRIMER-E Ltd, Plymouth, UK).

Results

Significant differences were found between unburned and burned plots, for almost all the variables included in this study. A total richness of 22 species was found for the study area, herbs were the most common (Table 1). We did not find pine regeneration in unburned plots, while in burned plots pine regeneration was present in all the plots. The density ranged between 0 and 46 individuals. Average number of pines for plots with regeneration was 11.1 (±12.8) individuals and biovolume 4.7 (±6.6) m^3. More than 40% of the individuals presented mature pine cones. In addition to *P. culminicola*, we also found another woody species, *Juniperus*

Table 1. List of the species recorded on the plots and Raunkaier classification.

Species	Raunkaier classification
Achillea millefolium	Camaephyte
Bromus carinatus	Cryptophyte
Campanula rotundifolia	Camaephyte
Ceanothus fendleri	Phanerophyte
Cirsium ehrenbergii	Camaephyte
Euphorbia beamanii	Camaephyte
Garrya ovata	Phanerophyte
Geranium crenatifolium	Camaephyte
Gnaphalium liebmannii	Camaephyte
Grindelia inuloides	Phanerophyte
Hymenoxys insignis	Camaephyte
Juniperus zanonii	Phanerophyte
Lupinus cacuminis	Camaephyte
Penstemon leonensis	Therophyte
Pinus culminicola	Phanerophyte
Polemonium pauciflorum	Therophyte
Ribes ciliatum	Phanerophyte
Salvia microphylla	Phanerophyte
Senecio coahuilensis	Camaephyte
Senecio loratifolius	Therophyte
Tauschia madrensis	Camaephyte
Villadia aristata	Cryptophyte

zanonii, regenerating in 5 of the burned plots and in one unburned plot. Density of this species ranged between 13 and 3 in burned plots and one individual in an unburned plot. Biovolume ranged between 0.03 and 0.01 m^3 in burned plots and 0.08 m^3 in unburned ones. No evidence of dead individuals after 1998 was found in burned or unburned plots. Richness did not differ significantly between unburned *vs*. burned plots, however, Smith & Wilson evenness index was significantly higher (pseudo F_{38}=2.95, p<0.05) in burned plots (Table 2).

The environmental variables and soil nutrient variables average per plot are indicated in Table 3. Only 5 of these variables are correlated with the DCA axis I site scores: Litter, rock, pH, Ca and Mg (Pearson correlation coefficient -0.81, 0.49, 0.64, 0.45 and -0.53; all of them significant for a p<0.01 after Holm correction). We found all of these variables were significantly different when comparing burned *vs*. unburned plots.

As expected, litter cover was higher in unburned plots (pseudo F_{38}=64.04, p<0.01), while rock cover higher in burned plots (pseudo F_{38}=24.04, p<0.01). The pH was higher in burned plots (Pseudo F_{38}=16.87, p<0.01). Mg was significantly higher in unburned plots (pseudo F_{38}=13.74, p<0.01), while Ca was higher in burned plots (pseudo F_{38}= 4.83, p<0.05).

The DCA analysis (eigenvalues of axes I and II were 3.41 and 3.29, respectively, and the cumulative percentage of variance explained by both axes was 33.3 %) discriminated significantly unburned and burned plots along axis I using the logistic regression analysis of the plot scores taking as dependent variable burned *vs*. unburned (B = -3.642, Wald = 19.119, p<0.01). Based on cover, *P. culminicola*, *Salvia microphylla* and *Tauschia madrensis* were the most representative of the species composition in unburned plots (solid line polygon, Fig. 2), while the shrub *Ribes ciliatum* and forbs *Hymenoxys insignis* or *Villadia aristata* were more representative for the species composition of burned plots (dashed line polygon, Fig. 2). The rest of the species have similar importance in plots, such as the shrub *Juniperus zanonii* or well-distributed species like *Achillea millefolium*, *Penstemon leonensis* or *Senecio loratifolius* (Fig. 2).

Discussion

Studies into forest fire effects have mainly concentrated on the analysis of the canopy structure, but there have been fewer on the effects on the understory

Table 2. Biotic characteristics (225 m^2) of the plots (average and standard deviation).

	Unburned	Burned
Regeneration density/plot *P. culminicola*	0	11.1 (12.8)
Regeneration density/plot *J. zanonii*	1	6.0 (4.2)
Biovolume m^3/plot *P. cuminicola*	0	4.7 (6.6)
Biovolume m^3/plot *J. zanonii*	0.08	0.3 (0.2)
Dead individual *P. culminicola* after 1998	0	0
Species richness	6.5 (1.4)	5.8 (1.2)
Smith & Wilson evenness	0.3 (0.1)	0.5 (0.2)

Table 3. Environmental and nutrient content variables measured in unburned plots *vs*. burned plots (mean and SD). The variables significantly correlated with the axis I of DCA analysis are indicated in bold (for these variables, significant differences between unburned *vs*. burned are indicated as $p<0.05$ as "*" and $p<0.01$ as "**").

	Unburned plots		Burned plots	
	Mean	**SD**	**Mean**	**SD**
Altitude, m	3629.47	31.64	3640.71	38.89
Slope, °Sex	33.42	8.28	35.29	5.53
Litter, %**	**85.79**	**10.88**	**27.97**	**15.28**
Rock, %**	**10.87**	**8.61**	**30.67**	**15.73**
pH**	**6.94**	**0.23**	**7.23**	**0.08**
OM, %	14.61	3.51	13.64	3.02
N, mg/g	7.55	1.90	6.59	2.12
P, ppm	44.18	15.95	45.66	28.37
K, ppm	345.80	123.71	302.82	125.54
Na, ppm	219.81	119.51	269.13	172.33
Mg, ppm**	**50.62**	**30.96**	**22.37**	**13.16**
Ca, ppm*	**3079.93**	**1352.52**	**4225.66**	**1544.48**
EC, S/cm	144.77	26.32	134.30	14.44
Sand, %	46.25	8.21	41.60	5.89
Clay, %	16.68	4.82	19.61	5.19
Lime, %	37.07	5.96	38.79	4.66

species composition (Arévalo *et al.,* 2001). In our study, multivariate analysis revealed some statistically significant patterns of species composition, and the bivariate analyses revealed low differences in species richness. Also we found differences in the structure of the pine stand and soil nutrient composition.

Although we did not find significant differences in species richness, evenness was significantly higher in the burned plots. A reduction of the dominance of *P. culminicola* explains these differences. It is more common to find an increase in species richness together with evenness after natural fires (Shafi & Yarranton, 1973; Chaturvedi & Rafhubanshi, 2014; Markham & Essery, 2015). In any case, in spite of the higher evenness of the burned plots, the values were very low and far from maximum values, indicating the dominance of a few species such as *P. culminicola*, *Senecio coahuilensis* and *T. madrensis* in unburned plots and *Ceanothus fendleri* and *Euphorbia beamanii* in burned plots.

Regeneration of woody species has been considered to be susceptible to climate conditions and particularly to extreme events, responding faster than adults to environmental changes or extreme climatic conditions (Lloret *et al.,* 2012). In our case, regeneration was present only in burned plots and not in unburned ones (only adult trees were present). Nevertheless, a low growth rate was apparent and recovery was still very low from the situation prior to the fire. Though there was a source of seeds close enough and almost all the burned plots were colonized by *P. culminicola*. However, in this environment, due to the short growing period and low temperatures, some constraints (Körner & Peláez, 1989; Häsler *et al.,* 1999) are expected on root or shoot growth. This alpine ecosystem also requires higher rates of respiration of assimilated CO_2, which can explain in part the low growth rate of species (Atkin *et al.,* 1996) together with large pools of non-structural carbon in field-growth alpine leaves (Mooney & Billings, 1961). As a result of the low recruitment and growth in alpine environments, *P. culminicola*, like other species, revealed a high survivorship rate (Forbis & Doak, 2004), with no dead individuals being found in the burned plots since the burning.

Species composition was significantly different between unburned *vs*. burned plots. Dominance of *P. culminicola* in burned plots did not reduce richness, but significantly reduced evenness (Fig. 2). Species as *S. microphylla* and *T. madrensis* were dominant in control plots, while the shrub *R. ciliatum* and the forbs *H. insignis* or *V. aristata* were dominant in burned plots. Evident environmental differences among plots explain these differences in species composition, such as the litter or rock cover. But nutrient level (higher Ca in the burned plots and Mg in unburned plots) is also related to these differences (Certini, 2005). So, more time is still required for the recovery of the area.

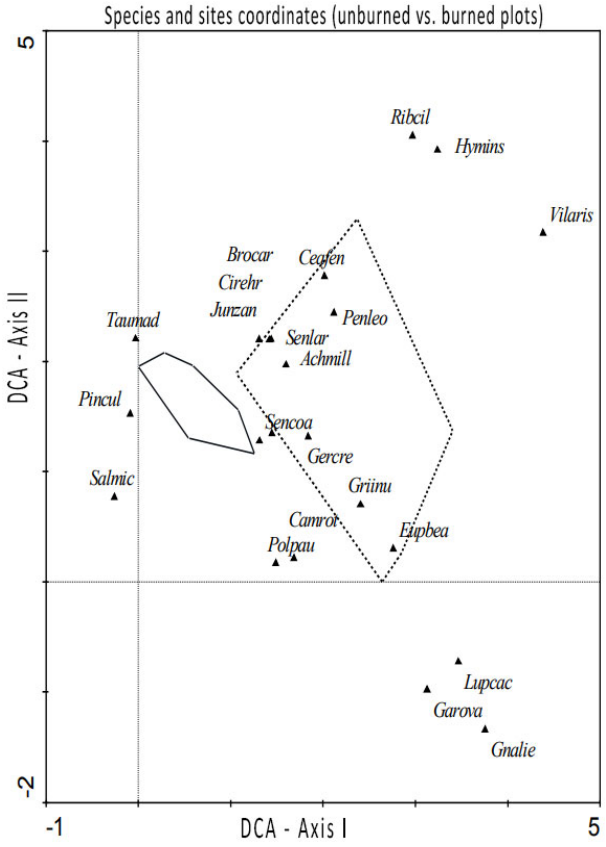

Figure 2. Species and plot scores in the space defined by axes I and II of DCA based on the matrix cover of the species. Dashed line polygon encloses burned plots; solid line polygon encloses unburned plots. The names of the species use the first three letters of the genus and the first three letters of the specific epithet (*Achillea millefolium, Bromus carinatus, Campanula rotundifolia, Ceanothus fendleri, Cirsium ehrenbergii, Euphorbia beamanii, Garrya ovata, Geranium crenatifolium, Gnaphalium liebmannii, Grindelia inuloides, Hymenoxys insignis, Juniperus zanonii, Lupinus cacuminis, Penstemon leonensis, Pinus culminicola, Polemonium pauciflorum, Ribes ciliatum, Salvia microphylla, Senecio coahuilensis, Senecio loratifolius, Tauschia madrensis, Villadia aristata*).

Fire is an intrinsic characteristic of this and similar ecosystems, and fires may be needed to maintain such systems (Arno & Allison-Bunnell, 2002), though it is also true that some studies have found negatives effects of fire on alpine plant communities (Kirkpatrick & Dickinson, 1984). Previous studies (García & González, 1991) did not report seedlings or saplings in burned areas of Cerro El Potosí during the field trips 13 years after the 1978 fire. In the case of the *P. culminicola* stand there is no information available about fire rates, but based on these results it is expected to take over 50 years for the plant community to recover (as a conservative estimate). Fires appear to favor natural regeneration (which is absent in unburned plots, with only adult trees present), and their impact on recurrence in vegetation recovery is well known (Tessler *et al.*, 2016), especially in these areas with such extreme environmental conditions. It is also clear that mountains areas are especially sensitive to human disturbances or changes in the natural disturbance rate (Guisan & Theurillat, 2000). Grazing is another factor that can limit recovery of plant cover in the area (Jiménez *et al.*, 2005) and is determinant in some ecosystems of the plant community (Arévalo *et al.*, 2007). Although we did not record of any grazing in the studied area, probably due to the steep slope of the burned and unburned area, there is some evidence around. In this case, grazing of cattle and horse should be prevented as it has been demonstrated that they are potential seedling grazers of this species (Jiménez *et al.,* 2005).

In conclusion, continuous monitoring of the affected area will be required to supervise the level of recovery and prevent external disturbances (as commented) to favor the establishment of natural regeneration of *P. culminicola*. However, we do not share the negative view about fire, as we consider it an intrinsic element of plant community dynamics. Nevertheless, conservation efforts are urgent to maintain a natural fire regime (recurrence, intensity, severity) and ensure the exclusion of grazing in order to favor the natural succession of the stand dominated by this unique species.

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